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A SYNOPSIS OF THE IMPACTS OF FOREST PRACTICES ON WATER QUALITY IN NORTH AMERICA

by
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A Synopsis of the Impacts of Forest Practices on Water Quality in North America

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Abstract

Forest management activities may substantially alter the quality of water draining forests, and are regulated as nonpoint sources of pollution. Important impacts have been documented in some cases for undesirable changes in stream temperature, and concentrations of dissolved oxygen, nitrate-N, and suspended sediments. We present a comprehensive summary of North American studies that have examined the impacts of forest practices on each of these parameters of water quality. In most cases, retention of forested buffer strips along streams prevents unacceptable increases in stream temperatures. Current practices do not typically involve addition of large quantities of fine organic material to streams, and depletion of streamwater oxygen is not a problem; however, sedimentation of gravel streambeds may reduce oxygen diffusion into spawning beds in some cases. Concentrations of nitrate-N typically increase substantially after forest harvesting and fertilization, but only a few cases have resulted in concentrations approaching the drinking-water standard of 10 mg of nitrate-N/L. Road construction and harvesting increase suspended sediment concentrations in streamwater, with highly variable results among regions in North America. The use of best management practices usually prevents unacceptable increases in sediment concentrations, but exceptionally large responses (especially in relation to intense storms) are not unusual.

Key terms: water quality, forest management, nonpoint source pollution

Stream water quality is a function of a variety of parameters, including temperature, sediment loads, inorganic chemistry, and toxic metals and organic compounds (EPA, 1986; MacDonald et al., 1991). Forest management practices, such as road construction, harvesting, and regeneration, may substantially alter the quality of water draining from forested watersheds. Degradation of water quality from forestry operations is considered a nonpoint source of pollution, regulated by an array of local, state, and federal regulations (Brown et al., in review). In some cases, forest practices have resulted in large or unacceptable changes in stream temperature, and concentrations of dissolved oxygen, nitrate-N, and suspended sediments. In this paper, we review the impacts of forest practices on these four parameters of water quality. This is a synopsis of a detailed cataloging of the effects of forest management on water quality based on experiments in more than 40 experimental forest areas in the United States and Canada (Binkley and Brown, 1992).

Temperature

Temperature affects both chemical and biological characteristics of streams. For example, the solubility of oxygen decreases rapidly as temperature increases. A change from 10 to 15 °C drops oxygen solubility by almost 20% (Golterman et al., 1978), and removal of tree canopies from over streams commonly raises stream temperatures by 3 to 7 °C (Brown, 1989). Most aquatic organisms have optimal temperature ranges; forest practices that change temperatures more than about 2 °C from natural temperatures may be enough to alter development and success of fish populations (Hornbeck et al., 1984).

Many of the early studies on the effects of forest harvesting on temperature did not include a leave strip of trees along streams to buffer temperatures, and such treatments typically allowed the maximum summer temperature to increase by 2 to 6 °C (Figure 1, Table 1). More

recent experiments, and most best management practices, have included buffer strips to moderate changes in stream temperature. In most cases, retention of buffer strips keeps the maximum increase in stream temperature to less than 2 °C.

Dissolved Oxygen

The concentration of oxygen dissolved in streamwater is important for fish and other components of aquatic ecosystems. Streams typically contain about 5 to 10 mg of oxygen/L, with lower concentrations for streams with high organic matter and high temperature. Streams containing spawning salmonid fish should not drop below 8 mg of O_2/L for one day, or below 9.5 mg/L for a 7-day mean; concentrations of 5 to 6.5 mg/L may be sufficient for adults (MacDonald et al., 1991). Only a few studies have examined changes in oxygen concentrations following forest harvesting. In Georgetown County, South Carolina, Askew and Williams (1986) examined the impacts of conversion of a 2400 ha watershed from natural hardwood forest to loblolly pine plantations. Dissolved oxygen increased marginally from the hardwood control forest (5.1 mg/L) to the older pine forest (6.9 mg/L). In Quebec, Plamondon et al. (1982) examined the effects of massive inputs of logging debris into a low-gradient stream (<1%). The logging debris impounded the stream, and lowered dissolved oxygen concentrations to near 0 mg/L. In the Coast Range of Oregon, Hall and Lantz (1969) found that accumulation of logging debris in a small stream depressed dissolved oxygen concentrations as low as 3 mg/L in summer (relative to a control stream with about 10 mg/L). The Caspar Creek Watershed Study in California (Rice et al., 1979) recorded depressions in dissolved oxygen concentrations in logged areas down to about 5 mg/L in summer, compared with an expected atmospheric equilibrium concentration of about 7 mg/L.

The input of fine organic debris following forest harvesting is usually low enough that

dissolved oxygen concentrations are not substantially depressed, especially where high turbulence replenishes oxygen from the atmosphere (Ice, 1978). However, forest practices that do not lower the concentration of dissolved oxygen in the water column may still lower the oxygen concentration in the streambed gravels, through addition of fine sediments that impede downward diffusion of oxygen (Everest et al. 1987, MacDonald et al., 1991). More work is needed to assess changes in oxygen concentrations within streambed materials following forest practices.

Nitrate-N

Forest practices may elevate the concentrations of many chemicals in streamwater, but only the concentrations of phosphate and nitrate are of significant concern in forestry (changes in concentrations of other nutrients, such as calcium and potassium, have little if any effect on aquatic ecosystems and the quality of water for designated beneficial uses). Phosphate concentrations in streams commonly limit the productivity of aquatic plants, and increases in P concentrations can lead to increased primary productivity and altered foodwebs in streams. Phosphate concentrations do not reach levels of concern for drinking water, but a standard of 0.1 µg/L has been set to prevent eutrophication of estuaries (MacDonald et al., 1991). No standard has been set for freshwaters because risk of eutrophication is very location dependent; a level of 0.05 mg/L may be sufficient to protect lakes (MacDonald et al., 1991). In a recent review, Salminen and Beschta (1991) concluded that changes in phosphate concentrations in streams after forest harvesting are uncommon unless accompanied by high-intensity slashburns, and that even in such cases the increases are short-lived. Forest practices do not appear to degrade water quality with regard to phosphate concentrations.

Nitrate-N concentrations above 10 mg/L are unacceptable for drinking water, because

of risks to infants (EPA, 1986). There is no generally accepted threshold of nitrate toxicity for aquatic ecosystems, but the eggs of some salmon species have shown sensitivity to levels of 10 mg-N/L (Kincheloe et al. 1979, Bisson et al. 1992). The drinking water standard is probably sufficient for protection of aquatic ecosystems (Norris et al. 1983). A variety of forest practices (such as harvesting and fertilization) often increase nitrate concentrations in streams, but concentrations are usually well within drinking water standards.

Harvesting

The effects of harvesting on streamwater concentrations of nitrate are plotted in Figure 2 from more than 30 studies in the U.S. and southern Canada (data in Table 2). Each point represents a pair of watersheds, one control and one harvested. Where more than one level of harvesting intensity was examined, Figure 2 includes the most severe treatment. Similarly, where the period of record after harvesting covers more than one year, Figure 2 lists the maximum year. About 70% of the studies found that average annual concentrations of nitrate remained below 0.5 mg-N/L for both control and harvested watersheds. Two patterns were apparent for the sites with greater concentrations. One group, primarily the forests of red alder and Douglas-fir in the Oregon Coast range, showed high levels of nitrate in both the control and harvested watersheds; no response to treatment was apparent. The other group, particularly the northern hardwood forests at Hubbard Brook, showed substantial nitrate increases after strip-cutting harvesting (Hornbeck et al., 1987). [Note: this Hubbard Brook example involves normal forest harvesting, *not* the inhibition of regrowth with herbicides which produced even greater nitrate-N losses; Likens et al. 1970.] In no cases did the average annual nitrate concentration exceed the drinking water standard of 10 mg-N/L; however, the alder/conifer watersheds and the harvested Hubbard Brook watersheds did have pulses of nitrate that

exceeded the standard.

How common are the high nitrate values for northeastern forests after logging? Martin et al. (1984) summarized the effects of conventional clearcutting practices on water quality in 38 watersheds from around New England. Vegetation types were described as conifer [white spruce (*Picea glauca*), red spruce (*Picea rubens*), black spruce (*Picea mariana*), and balsam fir (*Abies balamea*)], northern hardwood, and central hardwood [oaks, hickories, and red maple (*Acer rubrum*)]. Watersheds were not paired as controls and treatments, so these data are not included in Figure 2. Streamwater concentrations of nitrate for unharvested watersheds were near 0 mg-N/L for central hardwood forests, between 0.15 and 0.5 mg/L for the conifer forests, and 0.15 to 1.0 mg/L for northern hardwoods. Clearcutting from 20% to 100% of the watershed resulted in no nitrate increases in streamwater draining central hardwood or conifer forests. Streamwater nitrate-N concentrations for northern hardwood forests showed no effect of clearcutting if < 70% of the watershed was harvested, and either no change or an increase up to 2.0 mg/L in completely harvested watersheds. The authors concluded that their extensive survey across several vegetation and soil types found no cases where the increase in nitrate concentrations were as marked as those found after harvesting northern hardwoods forests at Hubbard Brook in the White Mountains of New Hampshire.

We conclude that harvesting does not degrade water quality through increasing nitrate concentrations in streamwater, with the possible exception of the Hubbard Brook forests. Even in the Hubbard Brook cases, rapid revegetation cuts the high nitrate losses, and downstream dilution also serves to reduce nitrate concentrations. However, high rates of atmospheric deposition of nitrogen compounds may already be generating high nitrate-N losses from some high-elevation forest in the eastern U.S. (such as the Smoky Mountains; Silsbee and Larson, 1982; Johnson and Lindberg, 1992), and current generalizations about low-nitrate losses from

eastern forests may not apply within several decades (cf. Aber et al., 1990; Johnson, 1991).

Fertilization

Most studies of the effects of fertilization on water quality come from the Pacific Northwest U.S. and British Columbia. The effects of fertilization on stream chemistry have varied substantially among studies. Fredriksen et al. (1975) tabulated the results from 6 studies. At Coyote Creek in southwest Oregon, fertilization with 225 kg urea-N/ha led to peak concentrations of nitrate-N (three days after fertilization) of about 0.17 mg/L. A second, similar peak in nitrate concentration developed several months later with the onset of the wet autumn season. The patterns were similar from another 28 studies, which showed peak nitrate-N concentrations of about 4 mg/L (EPA, 1980). Fredriksen et al. (1975) concluded that fertilization does not raises N concentrations to toxic levels, and poses no threat to stream water quality. Most studies (Figure 3, Table 3) support the general conclusion that N fertilization does not lead to excessive concentrations of nitrate-N in streams, with two exceptions. Hetherington (1985) examined streamwater nitrogen concentrations following fertilization with 225 kg-N/ha as urea on Vancouver Island, British Columbia; no effort was made to minimize application over streams within the unit. Post-fertilization concentrations peaked about 2 months after application, as heavy rains drained the soils. The maximum nitrate-N concentration reached 9.5 mg/L, very close to the 10 mg/L drinking water standard for the U.S. Nitrate concentrations returned to pretreatment levels during the first winter, with a second minor peak 1 year later. The second exception involves 2 fertilization experiments at the Fernow Experimental Forest in West Virginia. Kochenderfer and Aubertin (1975) reported peak nitrate-N concentrations of 16 mg/L in October following fertilization with 225 kg-N/ha as urea after clearcutting of Watershed 1. Helvey et al. (1989) and Edwards et al. (1991) examined the effects of fertilization with N (340 kg-N/ha as

ammonium nitrate) and P (100 kg-P/ha as triple superphosphate) on streamwater chemistry and nutrient yields. Fertilizer was applied by hand, with no direct application to streams. Streamwater nitrate-N concentrations exceeded the 10 mg/L drinking water standard for 3 weeks during the autumn after application.

Forest fertilization in the Pacific Northwest and British Columbia is unlikely to degrade water quality below drinking water standards (Bisson et al., 1992). The situation is less clear for forests in the East; the only site (Fernow Experimental Forest) examined clearly showed excessive nitrate concentrations in two separate trials.

Suspended Sediment

Increases in suspended sediment concentrations may degrade water quality for a variety of uses. High suspended sediment concentrations (or high turbidity) degrade the quality of drinking water. Increased suspended sediment concentrations may be associated with sedimentation of gravel streambeds, lowering permeability of the gravel beds and degrading habitat quality for spawning fish (MacDonald et al., 1991).

Natural variations in sediment concentrations can be very large, both within single streams and across regions of the country. Most streams draining forests have relatively low (<5 mg/L) annual average concentrations of sediments (Figure 4, Table 4), with stormflow peaks of up to 100 mg/L. However, a substantial number of sites have annual average concentrations that exceed 20 mg/L for undisturbed forests, with extreme outliers surpassing 1000 mg/L in at least some years. For example, an undisturbed forest of hardwoods and pine in Mississippi showed average annual concentrations of suspended sediment of 393 mg/L in one year, and over 2000 mg/L in another (Beasley, 1979). In Cherokee County, Texas, Blackburn et al. (1986) found that sediment concentrations ranged from 31 to 213 mg/L in an undisturbed pine forest.

Concentrations of suspended sediments often increase after management activities such as road construction, forest harvest, and site preparation. The increases can be highly variable, including some situations where no increases occur. The most representative example may be that of harvesting of a hardwood forest in the Leading Ridge watershed in Pennsylvania (Lynch et al., 1975; Lynch and Corbett, 1990). The use of Best Management Practices (BMPs) in the harvesting operation did not prevent an increase in sediment concentrations, but the increase was kept to a minimum (1.7 mg/L for the control watershed, 5.9 mg/L for the harvested watershed). In other cases, the use of BMPs may effectively prevent any increase in sediment concentration. For example, road construction, harvesting, and slash-burning in Fox Creek in the Bull Run Watershed in Oregon did not increase sediment concentrations beyond the 0.6-2.4 mg/L level of the control watershed (Fredriksen et al., 1975; Fredriksen and Harr, 1988).

In other cases, substantial (and variable) increases in sediment concentration occur. Harvesting of the hardwood/pine forest in Mississippi was followed by a variety of site preparation treatments (Beasley, 1979). In the first post-treatment year, annual average concentrations of suspended sediment rose from 2127 mg/L in the control to over 2800 mg/L for the more intensive site preparation treatments. In the second year, annual average sediment concentration declined to 393 mg/L in the control, but remained over 2300 mg/L for the most intensive treatment; concentrations for intermediate intensity treatments ranged between 600 and 900 mg/L. The causes of the high sediment concentrations include the intermittent nature of these small streams (flow stops for substantial periods between storms), the effect of forest removal on streamflow (increased volume and period of flow), and an apparent failure to retain a vegetated buffer strip next the streams.

Perhaps the major conclusions that can be drawn from the variety of studies in Figure 4 and Table 4 are that concentrations of suspended sediment differ substantially around the

country, and that the response to forest practices depends strongly on the particular location (including both site and weather conditions) and the details of the treatments that are applied. Similarly, the impacts of increased sediment concentrations on designated beneficial uses of water vary across regions. The design, utility, and efficiency of best management practices depends strongly on very local situations and conditions.

Conclusions

Forest practices have the potential to degrade the quality of water in streams by altering temperature, lowering dissolved oxygen concentrations, and increasing the concentrations of nitrate-N and suspended sediment. The retention of vegetated buffer strips protects streams from undesirable increases in temperatures in almost all cases (the one reported exception is Hewlett and Fortson, 1982). Depletion of streamwater oxygen is also rare in current harvesting operations; minimizing inputs of fine organic debris into streams prevents creation of high biological oxygen demand. Nitrate concentrations may be increased by forest harvesting, but any increase is well within acceptable water quality standards with a few notable exceptions (such as the northern hardwood forests at Hubbard Brook Experimental Forest). The major water quality concern in relation to forestry is suspended sediment, along with associated changes in channel conditions (such as the proportion of fine sediments in streambottom materials). The background concentrations of suspended sediments are variable across North America, as are the impacts of forest practices. From the research projects summarized here, the use of BMPs generally minimizes suspended sediment concentrations, though often at a substantial cost (Binkley and Brown 1992). More intensive monitoring of operational impacts of forest practices on sediment concentrations would be helpful for verifying the efficacy of BMPs in a wider range of less controlled situations.

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Table 1. Effect of harvest on temperature maximums.

State	Temperature (°C)		Measure	Overstory treatment[1]	Reference
	control	treat.			
East					
GA	21.1	25.0	Avg. daily for hottest month	CC, with buffer (12 m)	Hewlett and Fortson (1982)
NC	18.3	21.7	Avg. daily for hottest month	CC, no buffer	Swift and Messer (1971)
WV	14.4	16.1	Avg. weekly for growing season	95% CC, thin buffer	Aubertin and Patric (1974)
NH	16.0	20.0	Avg. daily for hottest month	100% CC, no buffer	Likens et al. (1970)
PA	19.4	20.6	Avg. daily for hottest month	44% CC, with buffer (30 m)	Rishel et al. (1982)
PA	17.8	25.0	Avg. daily for hottest month	85% CC, no buffer	Rishel et al. (1982)
West					
OR	14.4	15.0	A day in July	CC, with buffer	Brown et al. (1971)
OR	16.7	18.3	A day in July	CC, thin buffer	Brown et al. (1971)
OR	22.2	24.4	A day in July	CC, no buffer, reveg	Brown et al. (1971)
OR	13.3	15.6	A day in July	CC, no buffer	Brown et al. (1971)
OR	13.5	16.0	Avg. daily for hottest 3 weeks	25% CC, thin buffer	Harr and Fredriksen (1988)
OR	12.0	15.0	Avg. daily for hottest 3 weeks	25% CC, thin buffer	Harr and Fredriksen (1988)
OR	13.9	18.3	A day in July	CC, no buffer	Brown et al. (1971)
OR	14.4	18.9	A day in July	CC, no buffer	Brown et al. (1971)
OR	15.6	21.7	Avg. daily for hottest month	100% CC, no buffer	Levno and Rothacher (1967)
OR	16.1	23.3	A day in July	CC, no buffer	Brown et al. (1971)
OR	20.6	28.3	A day in July	CC, no buffer	Brown et al. (1971)
OR	14.4	22.8	Avg. daily for hottest month	100% CC, no buffer	Levno and Rothacher (1969)
OR	12.2	22.2	Avg. daily for hottest month	100% CC, no buffer	Brown and Krygier (1969)
BC	17.0	22.0	Avg. daily for July	60% CC, no buffer	Feller (1981)

[1] All overstory treatments were clearcuts of some or all of the watershed. Where the portion of the watershed that was clearcut was not provided, we listed the treatment simply as clearcut (CC).

Table 2. Effects of forest practices on streamwater nitrate concentrations (mg-N/L), as indexed by the maximum annual average nitrate concentration in post-treatment years of record.

State	Area	Concentration (mg/L)		Treatment	Reference
		Control	Treatment		
East					
FL	Bradford	0.03	0.04	80% clearcut	Riekirk (1983)
GA	Grant Forest	0.14	0.05	100% clearcut	Hewlett et al. (1984)
SC	Santee	0.1	0.1	Prescribe burned	Richter et al. (1982)
SC	Georgetown	0.5	0.9	Drained	Askew and Williams (1986)
NC	Coweeta	0.018	1.3	WS-7 clearcut	Swank (1988)
NH	Hubbard Brook	0.2	3.9	100% clearcut	Hornbeck et al. (1987)
PA	Leading Ridge	0.03	0.08	44% clearcut	Lynch et al. (1975)
WV	Fernow	0.2	0.2	100% clearcut	Aubertin and Patric (1974)
Central states					
TX	Cherokee Cty	0.01	0.3	100% clearcut	Blackburn et al. (1986)
NB	Nashwaak	0.12	0.6	100% clearcut	Krause (1982)
MN	Marcell	0.3	0.015	80% clearcut	Verry (1972)
West					
CO	Fraser	0.006	0.06	33% clearcut	Stottlemyer (1987)
ID	Silver Creek	0.01	0.018	23% clearcut	Clayton and Kennedy (1985)
ID	Priest River	0.71	1.51	Clearcut unit beside stream	Snyder (1975)
ID	Priest River	0.15	0.15	Clearcut unit beside stream	Snyder (1975)
ID	Priest River	0.015	0.18	Clearcut unit beside stream	Snyder (1975)
MT	Bitteroot	0.11	0.19	100% clearcut	Bateridge (1974)
MT	Bitteroot	0.17	0.13	100% clearcut	Bateridge (1974)
MT	Bitteroot	0.17	0.4	100% clearcut	Bateridge (1974)
UT	Chicken Creek	0.008	0.025	13% clearcut	Johnston (1984)
OR	HJ Andrews	0.08	0.07	WS-10 100% clearcut	Sollins and McCorrison (1981)
OR	Alsea	1.2	1.2	88% clearcut	Brown et al. (1973)
OR	Coast range	0.6	0.6	Herbicide/clearcut	Miller and Newton (1983)
OR	Coast range	1.7	1.7	100% Clearcut/herbicide	Miller and Newton (1983)
OR	Coast range	1.4	1.4	Herbicided	Miller and Newton (1983)
OR	Bull Run	0.01	0.08	25% clearcut/burned	Fredriksen et al. (1975)
OR	Coyote Creek	0.025	0.1	100% clearcut	Harr et al. (1979)
OR	High Ridge	0.003	0.162	100% clearcut	Tiedemann et al. (1988)
BC	UBC Research For.	0.04	0.5	100% clearcut	Feller and Kimmins (1984)
BC	Okanagan	0.03	0.03	16% clearcut	Hetherington (1976)
AZ	Beaver Creek	0.01	0.22	100% clearcut	M. Ryan, pers. comm.

Table 3. Effects of forest fertilization on maximum streamwater nitate-N concentrations (peak observation for control and post-fertilization periods, mg nitrate-N/L).

State	Area	Concentration (mg/L)		Treatment	Reference
		Control	Treatment		
East					
WV	Fernow Exp. Forest	0.2	16	225 kg-N/ha as urea after cc	Kochenderfer and Aubertin (1975)
WV	Fernow Exp. Forest	0.2	>10 for 3 wks	340 kg-N/ha as ammon. nitrate	Helvey et al. (1989)
West					
OR	Coyote Creek	0.005	0.18	225 kg-N/ha as urea	Fredriksen et al. (1975)
OR	Siuslaw Nat. Forest	0.5	2.1	225 kg-N/ha as urea	Fredriksen et al. (1975)
OR	Willamette Nat. Forest	0.1	0.13	225 kg-N/ha as urea	Fredriksen et al. (1975)
OR	Yamhill River	0.1	0.4	225 kg-N/ha as urea	Fredriksen et al. (1975)
OR	Siuslaw Nat. Forest	4.3	7.6	225 kg-N/ha as urea	USEPA (1980)
OR	Siuslaw Nat. Forest	0.16	0.13	225 kg-N/ha as urea	USEPA (1980)
WA	Olympic Nat. Forest	0.03	0.12	225 kg-N/ha as urea	Fredriksen et al. (1975)[1]
WA	Olympic Nat. Forest	0.005	0.04	225 kg-N/ha as urea	Fredriksen et al. (1975)[1]
WA	Louse Creek	0.1	3.0	200 kg-N/ha as urea	Bisson et al. (1992)
WA	Ludwig Creek	1.0	6.0	200 kg-N/ha as urea	Bisson et al. (1992)
WA	Various locations	<0.3	2.7	65 to 225 kg-N/ha as urea	Bisson (1982)
WA	Olympic Nat. Forest[2]	<0.01	0.07 to 0.72	225 kg-N/ha as urea	USEPA (1980)
WA	Entiat Exp. Forest	<0.1	0.15	110 kg-N/ha as urea[3]	Tiedemann et al. (1978)
BC	Mohun Lake	--	0.16	N fertilization with 50m buffer	Bisson et al. (1992)
BC	Mohun Lake	--	0.39	N fertilization with no buffer	Bisson et al. (1992)
BC	Lens Creek	0.05	9.5	225 kg-N/ha as urea	Hetherington (1985)
AK	Falls Creek	0.24	1.7	210 kg-N/ha as urea	Meehan et al. (1975)
AK	Three Lakes	0.20	2.7	210 kg-N/ha as urea	Meehan et al. (1975)

[1] Also USEPA (1980).

[2] 13 studies.

[3] Plus ammoniun sulfate.

Table 4. Effect of harvest and road construction on suspended sediment concentration.

State	Area	Concent. (mg/L)		Measure[1]	Treatment	Reference
		control	treat.			
East						
FL	Bradford Co.	3	4	2-year average	~80% clearcut, min impact	Riekirk (1983)
FL	Bradford Co.	3	13	2-year average	~80% clearcut, max impact	Riekirk (1983)
SC	Clemson E.F.	19	72	max. annual avg. in 3 yrs	Burn, 100% clearcut	Van Lear et al. (1985)
TN	Upper Coast Pl.	82	183	Max. stormflow in 3 years	100% clearcut	McClurkin et al. (1985)
MS	Upper Coast Pl.	2127	2471	Max. annual avg. in 2 years	100% clearcut, chopped	Beasley (1979)
MS	Upper Coast Pl.	2127	2837	Max. annual avg. in 2 years	100% clearcut, sheared	Beasley (1979)
MS	Upper Coast Pl.	2127	2808	Max. annual avg. in 2 years	100% clearcut, sheared	Beasley (1979)
PA	Leading Ridge	1.7	5.9	3-year average	Harvest 43%	Lynch et al. (1975)[2]
PA	Leading Ridge	1.7	80	2-year average	Clearcut, herbicide	Lynch et al. (1975)[2]
Central states						
TX	Cherokee Co.	79	36	Max. annual avg. in 4 years	100% clearcut, chopped	Blackburn et al. (1986)
TX	Cherokee Co.	112	1158	Max. annual avg. in 4 years	100% clearcut, sheared	Blackburn et al. (1986)
West						
ID	Priest River[6]	4.5	6.4	1-year average	Harvest	Snyder et al. (1975)
ID	Priest River[6]	7.1	37	1-year average	Harvest	Snyder et al. (1975)
ID	Priest River[6]	3	16	1-year average	Harvest	Snyder et al. (1975)
OR	HJ Andrews	9	240	6-year average	100% clearcut, burn	Fredriksen et al. (1975)
OR	HJ Andrews	9	430	6-year average	25% clearcut, road const.	Fredriksen et al. (1975)
OR	Santiam River[6]	71	50	9-year average	CC patches in large area	Sullivan (1985)
OR	Alsea	95	136	7-year average	road const, 25% CC, burn	Beschta (1978)
OR	Alsea	95	146	7-year average	road const, 82% CC, burn	Beschta (1978)
OR	Alsea	1	6	5-year average	25% clearcut	Fredriksen et al. (1975)[3]
OR	Alsea	1	10	5-year average	100% clearcut	Fredriksen et al. (1975)[3]
OR	Bull Run	2.4	2.4	Max. annual avg. in 8 years	Harvest 25%	Fredriksen et al. (1975)[4]
OR	Bull Run	2.4	2.4	Max. annual avg. in 10 yrs	Harvest 25%, burn	Fredriksen et al. (1975)[4]
OR	Coyote Creek	~20	~20	??	Shelterwood cut	Harr et al. (1979)[5]
OR	Coyote Creek	~20	~20	??	Patchcuts	Harr et al. (1979)[5]
OR	Coyote Creek	~20	170	??	Clearcut	Harr et al. (1979)[5]
WA	Hansel Creek[7]	3.7	178	Max. annual avg. in 3 years	Harvest, road construction	Fowler et al. (1988)
AK	Southeast	35	117	2-year average	Harvest, burn	Stednick et al. (1982)

[1] The years refered to are the initial post-treatment years. When "Max. annual avg." are reported for some time period, the concentrations reported here are for the year with the maximum concentration over the treatment watershed for the time period listed.

[2] Also Lynch and Corbett (1990).

[3] Also Brown et al. (1973).

[4] Also Fredriksen and Harr (1988).

[5] Also Adams and Stack (1989).

[6] A separate control watershed was not available. Control refers to upstream of the treatment area, and treatment to within or downstream of the treatment area.

[7] A separate control watershed was not available. Control refers to prior to treatment, and treatment refers to after treatment.

Figure Captions

Figure 1. Response of temperature maximums to forest harvesting at experimental watersheds (see Table 1 for details).

Figure 2. Response of streamwater annual nitrate-N concentrations to forest harvesting at experimental watersheds (see Table 2 for details).

Figure 3. Response of streamwater peak nitrate-N concentrations to fertilization (see Table 3 for details).

Figure 4. Response of suspended sediment concentrations to forest harvesting at experimental watersheds (see Table 4 for details)

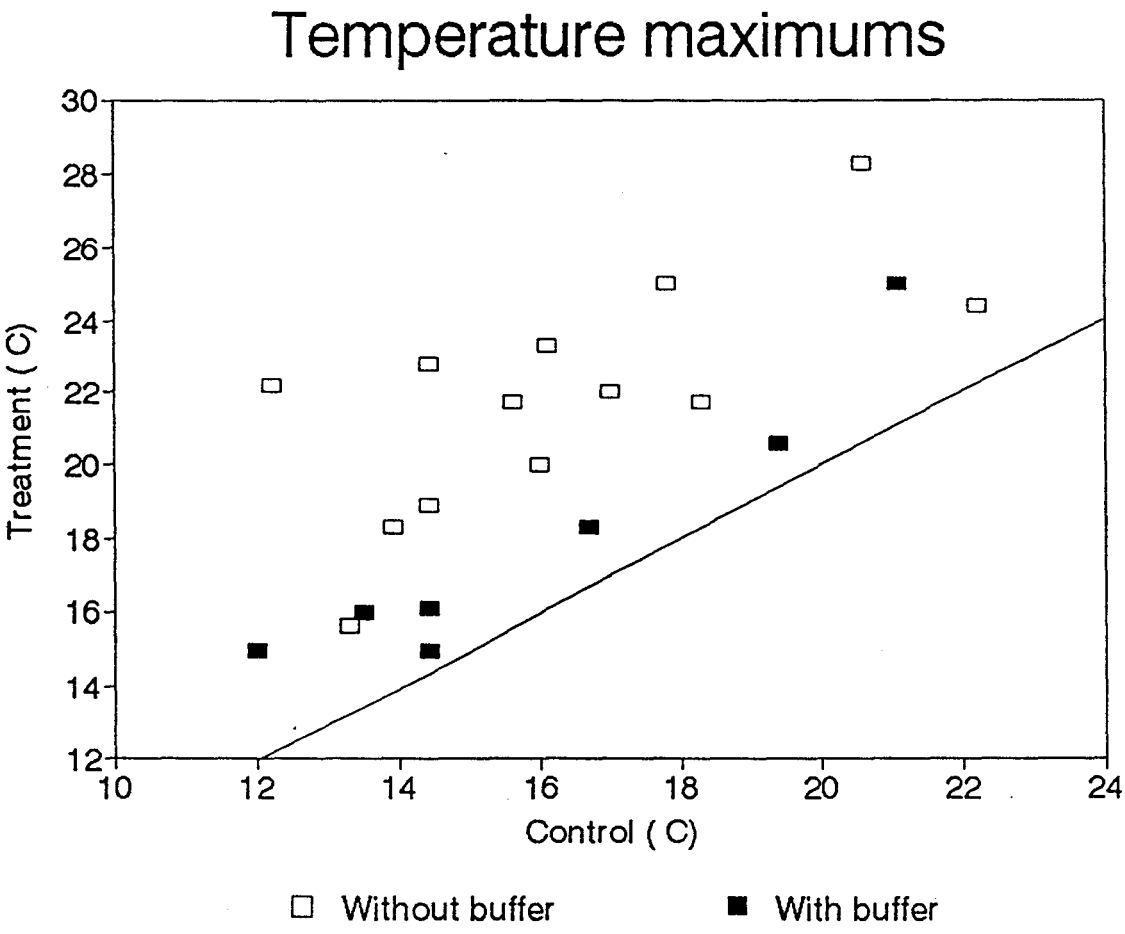
Temperature maximums
Treatment (C)
30
28
26
24
22
20
18
16
14
12
10
12
14
16
18
20
22
24
Control (C)
Without buffer
With buffer

Fig. 1

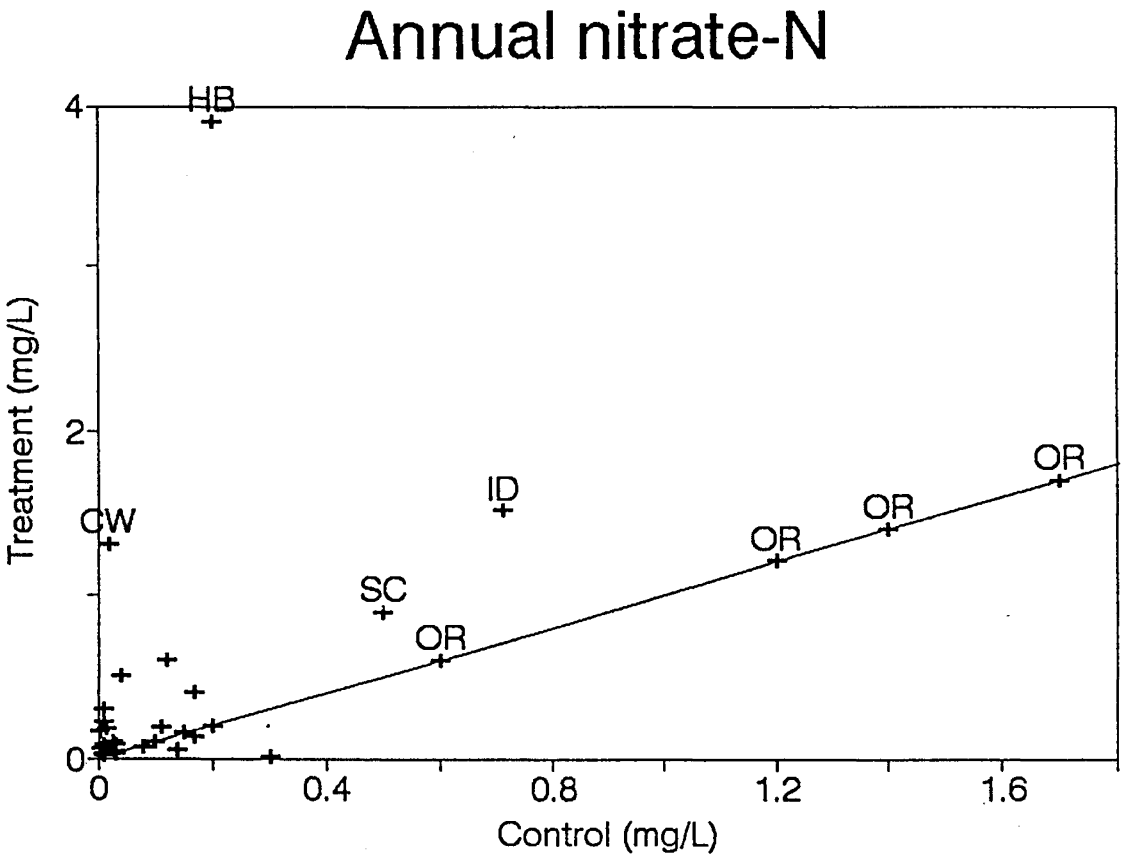
Annual nitrate-N
Treatment (mg/L)
Control (mg/L)
4
2
0
0
0.4
0.8
1.2
1.6
HB
CW
ID
SC
OR
OR
OR
OR

fig 2

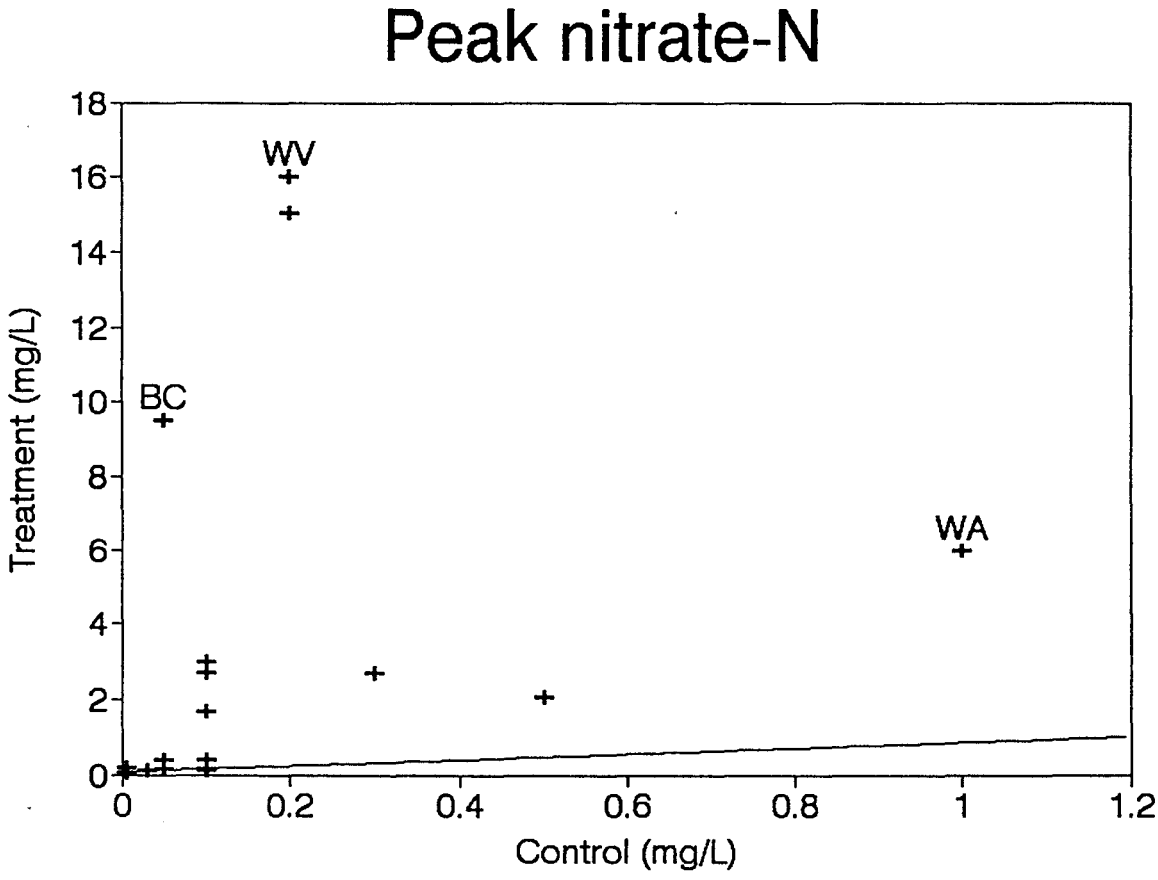
Peak nitrate-N
Treatment (mg/L)
Control (mg/L)
WV
BC
WA
0
2
4
6
8
10
12
14
16
18
0
0.2
0.4
0.6
0.8
1
1.2

fig. 3

Suspended sediment

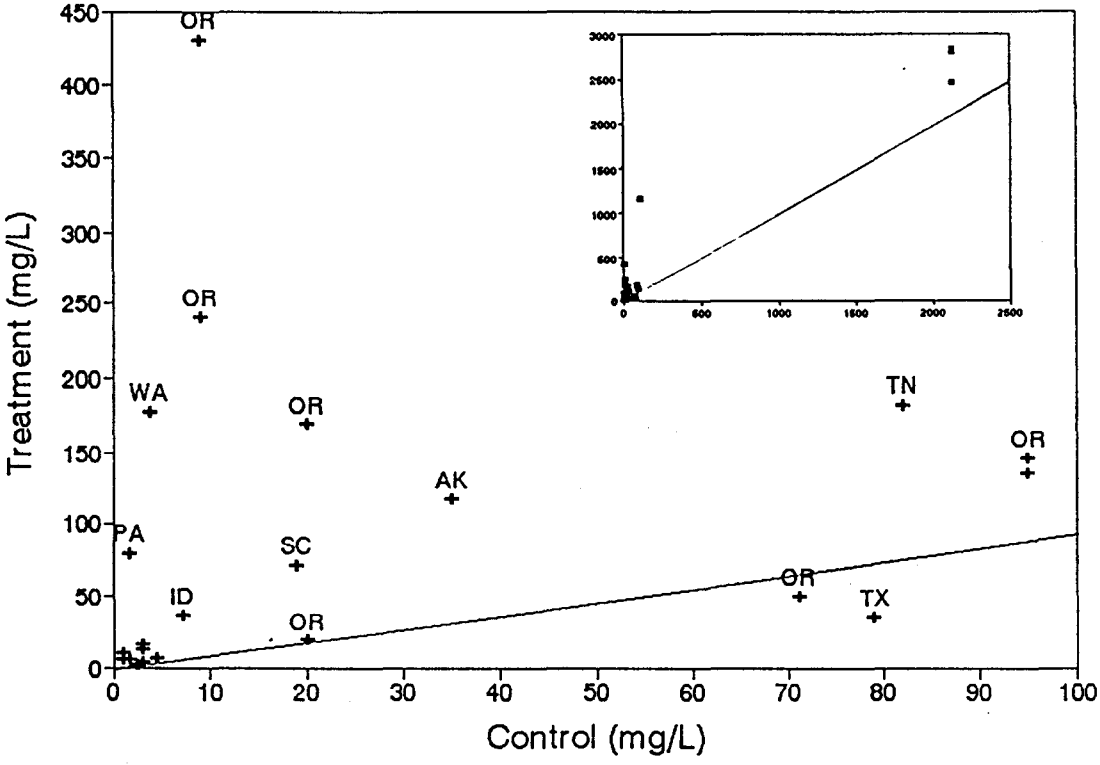

fig. 4